3 Research-based Principles Guiding Watershed Management

The purpose of this section is to identify various principles of watershed management that form the basis for the specific goals and implementation objectives for management of the Sudbury watershed lands during the period covered by the plan. These principles are distilled from a literature review of nearly 400 different sources, many of which are included in the Literature Cited listing at the back of this plan.

3.1 Principles of Watershed Protection

- Forested watersheds generally yield higher quality water than non-forested cover types. Urban, suburban and agricultural land uses all contribute in some way to lowered water quality.
- Uncontrolled human activities on water supply watersheds represent a major source of potential contamination. Efficient and effective water quality protection on both filtered and unfiltered water supplies requires control over human activities.
- Watershed cover conditions differ in their regulation of certain nutrients (especially phosphorus and nitrogen); the best regulation of nutrients is provided by vigorously growing forest that is fully occupying all watershed sites.
- Fire protection, police surveillance, water sampling, and other watershed management activities, including forest management, all depend upon an adequate, well-maintained road system.
- The proper management and protection of wetland and riparian zones is a critical component of watershed protection.

3.2 Principles of Watershed Forest Management: General

- Watershed forests can be managed in a way that provides significant benefits to long-term water quality protection, while minimizing adverse impacts during management operations.
- Potential negative tributary water quality effects (including turbidity, nutrients, and streamwater temperature) resulting from forest management can be minimized or eliminated with proper road location and maintenance and proper planning and supervision of silvicultural activities.
- Stands developed through uneven-aged methods will continually include some younger, shorter trees. Older trees in these stands develop stronger, more tapered stems than those grown in dense, even-aged stands. Strongly tapered trees sustain less damage from wind, and the younger component in uneven-aged stands enable them to recover from disturbance more quickly than maturing even-aged stands, thus improving their relative long-term water quality protection.
- Tree species growing on sites and within climatic ranges to which they are best adapted generally will grow vigorously and persist longer, resulting in a watershed forest that requires less tending. For example, white pine will grow vigorously but is more prone to root disease and wind throw on wet sites, while red maple tolerates soil saturation and remains wind firm on the same sites.

3.3 Principles of Watershed Forest Management Systems: Literature Review

3.3.1 Naturally-managed Forests

3.3.1.1 Water Yield

Tree growth and naturally occurring forest disturbances (fires, wind, disease, and insects) heavily influence the water yields from naturally-managed forests. Eschner and Satterlund (1965) studied a 491 square-mile watershed in the Adirondack Mountains of New York from 1912-1962. This study is particularly relevant to an examination of the impact of naturally-managed forests upon water yields. The land use on the watershed up to 1910 included land clearings, extensive fires, and heavy forest cuttings (chiefly logging of softwoods) that involved almost the entire watershed. In the late 1800s, the state of New York began purchasing lands in the watershed for the Adirondack Forest Preserve. From 1890 to 1910 the percentage of state-owned Forest Preserve in the watershed increased from 16% to 73%. The management policies of the Forest Preserve included laws against any cutting of trees and an active program of forest fire suppression.

The average forest density (in basal area) of the watershed increased from 65 square feet per acre in 1912 to 107 square feet per acre in 1952, due to forest growth and restrictions on cutting. Average basal area decreased to 97 square feet per acre in 1963 due in part to mortality from a windstorm in 1950. Another impact upon the watershed was a large increase in the beaver population during the study period. Throughout the Adirondacks, the number of beaver increased from an estimated 10 individuals in 1895 to an estimated 20,000 individuals by 1914, due to a prohibition on trapping introduced in 1895 and the introduction of 25 Canadian beaver and 14 Yellowstone Park beaver between 1901 and 1907. In 1965, most perennial drainages in the watershed had resident beaver.

The combined effects of unregulated forest growth and the increased number of beaver dams reduced the annual water yield of the watershed by 7.72 area-inches or 23% from 1912 to 1950. The authors postulated that forest growth reduced water yields through changes in evapotranspiration and snowmelt and beaver reduced yields through losses due to evaporation from beaver ponds. Although the net effect from beaver was a reduction in water yield, they tended to increase dormant season flow due to reduced interception and evapotranspiration following the killing of trees in flooded areas. Conversely, increased forest growth delays peak discharge and reduces yield. The effect of unregulated forest growth in lowering water yields was offset by increased in water yields resulting from the paving, straightening, and widening of 75 miles of roads within the watershed during the study period.

The trend of decreased water yields from 1912-1950 was reversed due to the large number of trees that were killed by the 1950 windstorm and the continued increase in mortality during the 13 years after the storm. The authors summarized the impact of the 1950 windstorm:

> The storm of November 1950, disrupted the associated patterns of forest stand development and streamflow change, returning both to a point nearer their 1912 levels.

In another study, Eschner (1978) analyzed four small watersheds in the Adirondack Mountains of New York. Logging, farming, and fires up to the early 1900s heavily impacted the East Branch of the Ausable River. Of the four watersheds, only the East Branch of the Ausable River was unaffected by the windstorm of 1950. Thus, this watershed offers a good example of a 42 year, stream-gauged period of uninterrupted forest re-growth. During this period, streamflow decreased by 4.2 area inches. Eschner concluded that this decrease was probably due to the natural regrowth of vegetation.

3.3.1.2 Water Quality

There have been few long-term studies of the impact of naturally-managed forests upon water quality. Several studies were cited above with regard to the impacts of old forests upon nutrient releases and the processes that are apparently involved. Other areas where naturally-managed forests may differ from actively-managed forests include response to natural disturbances, and nutrient/sediment interactions in stream channels.

The impact of disturbance is perhaps the key difference between a naturally-managed and actively-managed forest. In the actively-managed forest, silvicultural management is in effect a deliberate and regulated form of disturbance. In the naturally-managed forest, most disturbances are the result of unregulated natural events (e.g., wind, fire, disease, insects, or ice). While both actively-managed and naturally-managed forests will be exposed to certain recurring natural disturbances (e.g., hurricanes), the two systems may respond to these disturbances very differently.

In recent years, even forests isolated from developed areas are being increasingly impacted by human factors (air pollution, introduced insect/disease complexes, wildlife browsing). Eschner and Mader (1975) note:

> When extensive areas of relatively stable vegetation are set aside for wilderness, man's activities are sharply restricted. However, changes in the vegetation continue, and in some cases the possibility of catastrophic change increases...Treatment of large areas of watershed as wilderness, currently advocated by several interest groups, may not be consonant with management for maximum yields or protection of areas. On land long undisturbed, use of water by vegetation may be maximized and water yield reduced, while hazards of windthrow, insect, disease, or fire damage may increase.

Hewlett and Nutter (1969), in defining pollution, mention the potential impact of natural disturbances upon water quality:

> Because natural waters already carry materials that can degrade water for certain uses, we have some difficulty specifying just what "pollution" is. Natural water quality over the centuries has evolved the stream ecosystem under conditions that we might, rather pointlessly, refer to as "natural pollution." For our purposes, however, we shall regard pollution as man-caused and think of polluted waters as those degraded below the natural level by some activity of man. In this sense, therefore, unabused forests and wildlands do not produce polluted waters, although they may at times produce water of impaired quality.

Parsons et al. (1994), in a study of the impact of gap size on extractable soil nitrate stated:

> Large-scale mortality events leading to macroscale gap formation, which involves the simultaneous death of many adjacent trees over thousands or tens of thousands of square meters, are known to increase mineralization and nitrification rates in temperate forest ecosystems.

Tamm (1991), in reviewing the role of nitrogen in terrestrial ecosystems, noted:

> Natural agents such as storms, insect defoliations, and, above all, fire may destroy the

existing vegetation and stimulate both nitrogen mineralization and nitrification, leading to temporary losses of nitrate.

Corbett and Spencer (1975) report that Hurricane Agnes and the 14 inches of rain that accompanied it caused significant erosional impacts to the Baltimore Municipal Watershed, chiefly due to streambank cutting and channel slumping. The authors note that these types of impacts are more related to channel depth than condition of forest cover. Hurricane Hugo caused extensive damage to coastal South Carolina. The U.S.D.A. Southeast Forest Experiment Station monitored stream waters within the Frances Marion National Forest before and after the hurricane, with a gap in monitoring for several months after the hurricane, due to access problems (McKee, 1993, pers. comm.). The forest before the storm was mature pine-hardwoods and much of it was windthrown or snapped by the storm. Preliminary results show increased nitrogen in streams compared with levels found in regular monitoring done before the storm (Swank, Harms, Neary, Benston, McKee, and Hanson, 1990, 1991, pers. comm.).

Researchers in South Carolina are also concerned about the threat of a large forest fire due to the amount of downed material that has increased from 8 tons/acre before the storm to 100 tons/acre after the storm. After a 1.6 acre simulated hurricane "pulldown" at the Harvard Forest, Carlson (1994) reported that downed woody debris increased from 4.1 tons/hectare in a control area to 33.5 tons/hectare. He suggests that the potential threat of fire will increase in the next several years as pulled-over trees die.

Numerous studies show that impacts from forest blowdown or a combination of blowdown and forest fire can increase tributary nitrate and phosphorus exports by several times background levels (Verry, 1986 and Packer, 1967 as cited in Ottenheimer, 1992; McColl and Grigal, 1975: Wright, 1976; Schindler et al., 1979). Soil disturbance from blowdown of large numbers of trees may also result in significant erosion (Patric, 1984, White et al., 1980, and Swanson, 1982, all cited in Ottenheimer, 1992). Water quality changes associated with extensive windthrow and fire confirm that dissolved nutrients and in some cases, sediment, acidity, and total organic carbon can be elevated for several years (Patric, 1984 and Swanson, 1982 as cited in Ottenheimer, 1992; Verry, 1986; Schindler et al., 1980; Wright, 1976; Corbett and Spencer, 1975; McColl and Grigal, 1975; Dobson et al., 1990; Dyrness, 1965 and McKee 1993, pers. comm.). For example, nitrates increased by up to nine times and phosphorus by more than three times after extensive windthrow followed the next year by a wildfire in a monitored watershed in Ontario (Schindler et al., 1980).

Dobson et al. (1990), in reviewing data from hundreds of lakes in New York, New Hampshire and Sweden, found strong spatial and temporal associations between percentage of watersheds affected by large blowdown events and long-term lowered pH in basin lakes. They concluded that extensive blowdown alters hydrologic pathways by channeling flow through large macropores created by rotting roots so that water is less buffered by subsurface soils and bedrock. One lake adjacent to heavy blowdown that was extensively salvaged did not acidify, leading the authors to speculate that salvage may partially counter the impacts of blowdown on acidification.

The value of advance regeneration (regeneration established before overstory removal) in reducing the impacts of natural disturbances may be the critical factor distinguishing actively managed and naturally managed watersheds. After disturbance, areas that are quickly occupied with dense, fast-growing seedling/sapling growth will minimize transitional losses of nutrients, and particulate and erosional losses. OWM foresters Buzzell (1991) and Kyker-Snowman (1989a) compared actively managed and naturally managed forests with regard to the presence and abundance of advance regeneration. Their findings definitively show that areas that have been actively managed have a much greater amount and density of regeneration and young forest growth. Arbogast (1957) also notes that a key consideration when implementing uneven-aged silviculture on previously unmanaged and undisturbed stands is to enhance age-class balance by encouraging development of sapling and pole-sized trees.

The impact of actively managed and naturally managed forests adjacent to stream channels is discussed thoroughly in Maser et al. (1988). Although this study is focused on forests of the Pacific Northwest, some principles are applicable to the northeast. The authors documented that streams flowing through young forests and those recently harvested contain only 5-20% of the large woody material found in streams flowing through naturally managed forests. The stability and length of wood pieces is also increased in naturally managed forests. While the authors document a clear difference in the fish habitat of the two streams, they also note that the increased debris in streams bounded by naturally managed forests may impact the stability of streams.

While it may seem that large amounts of woody debris would increase the amount of decomposed material in streams, wood in direct contact with water decomposes very slowly. The authors note that only 5-10% of a stream's nitrogen supply is derived from rotting instream debris. On the positive side, debris serves to create hundreds of dams that slow the flow of particulate material down the stream. The authors speculate that stream stabilization after floods is accelerated by large woody debris, noting that "large stable tree stems lying along contours reduce erosion by forming a barrier to downhill soil movement."

While the conditions in the Pacific Northwest are very different from those in the northeast (for example Pacific NW soils are less stable, forest types are totally different, and forest management systems consist generally of even-aged management using clear-cutting), some of the above material is applicable to the northeast and to OWM watersheds. Bormann et al. (1969), in a study of a small watershed in the White Mountains of New Hampshire noted that 1.4% of the watershed was included in the actual stream channel and that debris pools occurred every 1-3 meters. They speculated that these pools served to slow the movement of suspended material from the watershed and reduce the erodibility of the system. Bormann et al. (1974) note that in mature forests the export of particulate material is derived from material stored in the stream bed. However, they note that most of this material moves very little, and approximately 90% decomposes slowly in place.

The above discussion highlights the need for careful consideration of lands adjacent to tributaries. In developing management plans for these areas, consideration should be given to the need for stability of the cover type and forest structure, given the potential occurrence of major disturbances. However, the benefits of the slow addition of natural wood-fall to these areas, and the erosion impediments and the stream pools created by this material, should also be analyzed. In assessing the management of stream buffers, Stone (1973) recommends careful thinning of buffer strips as often preferable to complete non-disturbance, as such thinning will limit the amount of debris falling directly into streams. Vellidis (1994) found that forested riparian strips next to agricultural lands took up and removed nutrients in soil and vegetation, preventing agricultural outputs from reaching streams. The author recommends that these forested strips be harvested periodically to ensure a net active uptake of nutrients, if they are to serve as an effective nutrient buffer.

3.3.2 Even-Aged Silviculture

3.3.2.1 Water Yields

Beginning at Wagon Wheel Gap in Colorado in 1911, experiments relating forest removals to water yield increases have been conducted at a number of small watershed locations throughout the U.S. Since 1940, three U.S. Forest Service Experimental Forests have supplied the bulk of the data for eastern

U.S. applications. These forests are Hubbard Brook, NH; Fernow, WV; and Coweeta, NC. Experiments have included a wide variety of approaches ranging from clearing of small watersheds to patch, partial, and riparian cuts. Most experiments are paired watershed studies, where two small, adjacent or similar watersheds are studied; one watershed is treated silviculturally while the other is left intact, as a control.

Experimental findings show several general trends. However, variation due to site conditions such as slope, aspect, soils, geology, cover type, and additional factors make exact prediction of water yield increases difficult for a given site. Douglas (1983) notes that yield increases can be predicted within 14% of actual values. Federer and Lash (1978) developed a small watershed computer model aimed specifically at predicting water yield increases from forest management of small watersheds in the northeast, using input variables of precipitation, temperature, latitude, slope, aspect, cover type, and soils. This model was applied with a reasonable degree of accuracy to the Cadwell Creek watershed at Quabbin (O'Connor, 1982b).

The following general trends emerge from the many watershed experiments that have been reviewed for the development of watershed land management plans:

- Water yields increase as the percentage of forest cover removed increases - complete removal of hardwood cover on small watersheds can result in first-year yield increases of 4-14 area-inches (total average annual streamflow in the Northeast is approximately 20-25 area-inches or about 50% of total precipitation).
- Water yields decrease with reforestation of open watersheds and growth of younger forests, with a linear relationship between the percentage of watershed reforested and water yield decrease; yield decreases are significant, in the range of 6-7 area-inches lost through significant forest regrowth and forest growth.
- Water yield increases are greatest the first year after cutting and decline thereafter, usually returning to pre-cutting levels by the 4th to 8th year; most clearing experiments returning to pre-cut levels within 10 years.
- Water yield increases are generally larger on north versus south facing slopes, with yields up to two and one half times greater for clearings on north facing slopes. One study also showed that west-facing forests used more water than those did on east-facing slopes.
- Differences between cut and uncut watershed yields increase exponentially as annual rainfall increases.
- Water yield increases from cutting in the many studies in the northeast occurred chiefly during the growing season, with areas of higher snowfall, deep soils, or conifer cover showing larger dormant season increases.
- Removal of conifer forests will yield more water than hardwood forests, as conifers use more water and snow evaporation is greater in conifers.
- Conversion of hardwoods to conifers will result in significant losses in water yields - one watershed in North Carolina had a 25% yield loss after conversion of hardwoods to white pines.
- Greatest yields are usually achieved through removal of riparian vegetation or lower elevation watershed vegetation.
- Much of the increased flow generated from cutting is seen as increases in low flow periods. Increases in peak flows do occur, but are not believed to cause increased flood risk where cutting is implemented on limited areas and moderate increases are generally yielded.

- Watersheds with deep soils generate longer lasting flow increases after cutting, and yields are more balanced between growing and dormant seasons; watersheds with shallow soils generate yield increases focused within the growing season.
- Certain early successional hardwoods use measurably more water than late successional hardwoods, and changes in water yield due to shifts in species composition may last in excess of a decade.
- Yield increases are lower in deep soils and in areas with fast regrowth of regeneration.

(Douglass and Swank, 1972, 1975; Douglass, 1983; Hibbert, 1967; Federer and Lash, 1978; Hornbeck and Federer, 1975; Hornbeck et al., 1993; Lull and Reinhart, 1967; Mader et al., 1972; More and Soper, 1990; Mrazik et al., 1980; Storey and Reigner, 1970; Trimble et al., 1974.)

Douglass (1983) and Storey and Reigner (1970) emphasize the significance of the above findings as a way to help meet present and future water supply needs in the eastern United States. Given the above summary, the types of management that will yield the most water are those consistent with even-aged management, especially involving large clear cuts.

While clear cutting of entire reservoir watersheds is not feasible for water quality reasons (see next section on water quality), judicious rotation of clear cuts may provide significant flow increases, especially during the growing season when they are most needed by water supply managers. Douglas and Swank (1972) summarize the value of forestry for water supply managers:

> We can conclude from the experimental watershed evidence in the Appalachian Highlands that cutting forest vegetation has a favorable impact on the water resource by supplementing man's supply of fresh water when consumptive demands are most critical. And, the amount of extra water produced can be predicted with a degree of accuracy that is sufficient for many purposes. Although heavy forest cuttings will usually increase some stormflow characteristics on that portion of the watershed cut over, regulated cutting on upstream forest land will not produce serious flood problems downstream.

3.3.2.2 Water Quality

In describing the impacts of even-aged and uneven-aged management upon water quality, most studies reviewed involved either clear cutting (of whole watersheds or in limited blocks or strips - all of which fall under even-aged management) or partial cutting (where part of or most of the overstory is retained). It should be noted that while partial cutting falls under uneven-aged management, variations of the shelterwood cutting system (a form of even-aged management involving removal of the forest overstory in stages) involve only partial cuttings.

The impacts of even-aged management systems upon water quality vary with intensity and location of management; intensity, layout and maintenance of road systems; and planning and supervision of logging and woods roads operations (Lull and Reinhart, 1967; Kochenderfer and Aubertin, 1975; Hornbeck and Federer, 1975). The water quality parameters principally affected by these activities are turbidity, nutrient levels, and stream temperature.

Turbidity

Turbidity is affected by soil exposed in poorly planned, located, and maintained road systems and log landings (Kochenderfer and Aubertin, 1975). For example, gravel access roads may have an infiltration capacity of .5 inches per hour, while forests have capacities of 50 inches per hour (Patric, 1977, 1978). Haphazardly built road systems may utilize 20% of a watershed, while well planned road systems may utilize 10% (Lull and Reinhart, 1967). In addition to access and skid roads, the total compacted area of a typical logging area including landings may approach 40% (Lull and Reinhart, 1972). MDC conducted a study in 1986 of pine thinning on the Wachusett Reservoir watershed completed by agency crewmembers and two private loggers under MDC supervision. For this study, the total area impacted by logging - including access roads, skid roads, and landings - ranged from 14.8% (MDC crew) to 19.6% (private loggers) (Kyker-Snowman, 1989b). Stone (1973) reported soil disturbances covering 15.5% of the logged area for selection cutting, versus 29.4% for clear cutting in eastern Washington. Sediment export was directly proportional to the percentage of a watershed in roads and reducing this percentage was seen as critical for reducing sediment in streams in the Pacific Northwest (Dyrness, 1965).

Hornbeck et al. (1986) report that increases in soil disturbance means greater erosion. Martin (1988) recommends setting predetermined travel routes for equipment and doing winter logging and using tracked vehicles rather than wheeled vehicles in sensitive areas. Hewlett (1978) recommends avoiding road locations near perennial and intermittent stream channels in order to eliminate impacts.

A study of erosion on New York City's water supply watersheds highlights the importance of protecting road and stream banks from the effects of erosion. This study of the erosion sources at the Schoharie Reservoir estimated that while road banks made up only .22% of the watershed, they were the source of 11% of all erosion. Streambanks, which made up only .44% of the watershed, were the source of 21% of all erosion (S.U.N.Y., 1981).

Construction of new access roads carries the greatest risk of erosion. Massie and Bubenzer (1974 as cited in O'Connor, 1982a) found that 36% of all road erosion in the study area was produced by roads two years old or less, although this category of roads made up significantly less than 36% of all roads. Stone (1973) notes that some turbidity is inevitable with construction and initial use of new roads, but that almost all continuing damage from roads is avoidable by using recommended woods roads maintenance techniques.

A comparison study of graveled and ungraveled forest access roads in West Virginia showed that the application of even 3 inches of gravel reduced sediment losses eight-fold, even though the gravel road carried two times the traffic of the ungraveled road (Kochenderfer and Helvey, 1974).

Lynch et al. (1975) traced increased turbidity on watersheds in Pennsylvania to scarified log landing areas. However, Kochenderfer and Aubertin (1975) report that:

> Bare soil exposed by road building, and to a much lesser extent by log landings, has long been recognized as the major source of stream sediment associated with logging operations.

Turbidity in a West Virginia watershed that was clearcut was traced to both road erosion and channel scour from heavier overland flow (Patric, 1976). Channel scour is an impact that is unique to large-scale clearcuts or disturbances where peak flows may increase.

Mechanical compaction of soil reduces soil infiltration and reduces tree seedling survival (Martin, 1988). Erosion problems result when mineral soil is exposed to rain, especially on areas with long, steep slopes. However, even compacted, exposed soils have high infiltration capacities. The most significant erosion occurs when soil is bared to the "B" horizon, beneath the organic and leached horizons (Patric, 1977).

MDC measured soil bulk density (a parameter which shows soil compaction) on transects through a pine thinning at Wachusett Reservoir with three types of conventional logging equipment. Average soil bulk densities did not change significantly when measured before and after logging done by MDC's crew with a conventional small skidder and a forwarder. Average bulk density before logging was 6.18 grams/cubic centimeter (gms/cm) and 6.21 after logging; 13 gms/cm is considered the level at which root penetration is inhibited. Stone (1973) reported that soil compaction varies enormously with soil type, moisture content, frequency of traffic, and type of "packing" impact. He concluded that the key to avoiding erosion from logging is to ensure that protection steps will handle extreme rain events on the most sensitive sites. The careful planning of skid roads is essential.

Cuttings where soils are not disturbed by roads or skidding do not discernibly increase turbidity (Kochenderfer and Aubertin, 1975; Dyrness, 1965; Bormann et al., 1974). In Connecticut, 80 logging locations were checked for compaction, erosion, and stream sedimentation. All such problems were found to be related to the transportation aspects of logging (O'Haryre, 1980, as cited in More and Soper, 1990). Other studies trace turbidity to erosion from heavily used logging roads, particularly after heavy rainstorms and from increased streamflow that caused channel erosion (Patric, 1976; Pierce et al., 1970 as cited in More and Soper, 1990).

Turbidity measurements were compared on watersheds in the Fernow Experimental Forest, West Virginia; treatments included a commercial clearcut, a silvicultural clearcut, and one watershed with no cutting. Turbidity (in Jackson Turbidity Units - JTU) during logging was 490, 6, and 2 units respectively. One year after cutting, turbidity was 38, 5, and 2 units respectively (Kochenderfer and Aubertin, 1975). Douglass and Swank (1975) concluded that well-planned, well-maintained road systems do not damage water resources. In a comparison of logging with planned and unplanned skid trails, the planned logging had turbidity of 25 JTU while the unplanned logging had 56,000 JTU (Reinhart and Eschner, 1962, as cited in Brown, 1976). A comparison of regulated and unregulated logging in 1947-8 found that unregulated logging increased turbidity 10-20 times background levels while regulated logging increased turbidity only slightly (Douglass and Swank, 1975).

In a study at Hubbard Brook, New Hampshire, a watershed was logged with a strip cut even-aged method. In the two years during and after logging, 6 of 147 streamwater samples exceeded 10 turbidity units (Hornbeck and Federer, 1975). A study of different stream crossing techniques in the Ware River watershed found that temporary bridge crossings caused less impact than ford crossings or crossing on poles. Increases in turbidity caused by temporary bridge crossings were not measurable beyond 100 feet downstream from the bridge (Thompson and Kyker-Snowman, 1989).

Clearing of riparian areas has been associated with increased turbidity (Corbett and Spencer, 1975). Lynch et al. (1975) compared middle slope clear cuts with lower slope clear cuts and found turbidities of 4 part per million (ppm) on middle slope cutting, 196 ppm on lower slopes, and 2 ppm on an uncut control watershed.

While useful predictive models exist to estimate soil loss from agricultural practices, few soil loss predictive models exist for silvicultural operations. Burns and Hewlett (1983) developed a model that evaluated clearcut, disking, and planting operations in the southeastern U.S. This model is based on the percentage of bare soil after logging practices and the location of bare soil areas with regard to perennial stream channels. The authors recommend keeping any exposed soil areas away from wet and dry stream channels in order to minimize erosion. Currier et al. (1979) developed a procedure for analyzing water quality impacts from forest management. Larson et al. (1979) began assembling existing data into a system of computer models. Li et al. (1979) developed a sediment yield model based on the Universal Soil Loss Equation and tested in Colorado.

Nutrients

Logging impacts on nutrient levels can vary by the amount of cover removed, type of cover removed, watershed slope, location within the watershed (lower areas cause faster nutrient input, but higher areas cause more nutrient loss), and the timing of the regeneration response. Soil type and depth also control impacts (e.g., deep, poorly-drained, fine-textured soils tended to bind free nutrients before they reached the streams) (Bormann et al., 1968; Brown, 1976; Carlton, 1990; Martin and Pierce, 1980; Martin et al., 1984). While turbidity increases are caused by soil disturbance, increases in nutrient levels can result solely from cover removal. For example, at Hubbard Brook, New Hampshire, all trees on a catchment were cut and left on the ground and herbicides applied to prevent regrowth. As a result, stream concentrations of several ions increased significantly (Douglass and Swank, 1972). In this study, nitrates increased more than forty times background amounts (Bormann et al., 1968). Cuttings associated with significant nutrient increases typically involve clearing of large percentages of watersheds. However, even clearing of entire watersheds at Fernow Experimental Forest, WV and Pennsylvania State Experimental Watersheds did not appreciably increase nitrates (Kochenderfer and Aubertin, 1975; Lynch et al., 1975).

Nutrient increases from cleared areas are derived both from the increases of nutrients released as the decomposition process increases in sunlight and by the reduction in uptake due to the loss of plant cover (Vitousek, 1985). Strip cutting of one third of a watershed (at Hubbard Brook, New Hampshire) caused nitrate increases of nearly two times an undisturbed watershed and one third that caused by a watershed that was completely clear-cut (Hornbeck et al., 1975). The coarse-textured soils of New England that have lower nutrient-holding ability may be more susceptible to nutrient losses, particularly in areas without plant cover (Hornbeck and Federer, 1975). Soils that are shallow to bedrock, thin unincorporated humus on infertile soil, and coarse skeletal soil on steep slopes are all also susceptible to nutrient loss (Williams and Mace, 1974). In areas where soils may be sensitive to nutrient loss, limiting cutting to light partial cuts may be necessary to prevent nutrient loss (Brown, 1976).

Aber et al. (1978) modeled changes in forest floor biomass and nitrogen cycling using various regimes of clear-cutting. A projected rotation that clear-cuts a forest each 30 years versus one on a 90 year cycle will accumulate less floor biomass and release more nitrogen to streams. Williams and Mace (1974) state that, in general, the more drastic the manipulation of the forest, the larger the corresponding release of nutrients, with minor manipulations causing little or no nutrient release. In their study of jack pine clear-cutting in Minnesota, summer logging involving whole tree removal was found to cause significantly more nutrient leaching than winter logging with only stem removal.

Temperature

Stream temperature is important in protecting aquatic life because of its impact on dissolved oxygen and nutrients (Brown, 1976). Stream temperatures vary depending on the presence of forested buffer strips adjacent to stream channels (Hornbeck et al., 1986). Douglass and Swank (1975) concluded, "stream temperatures are not increased by forest cuttings if a buffer strip is retained to shade the stream."

Kochenderfer and Aubertin (1975) found that clear-cuts on upper watershed areas did not increase stream temperature, as few stream channels occur in these areas. In lower watershed cuttings where trees were left adjacent to the stream channel, cuttings had no influence on stream temperature.

Summary

Studies indicate that erodibility of a watershed impacted by either natural disturbances or logging will remain low "as long as destruction does not involve severe and widespread disruption of the forest floor" (Bormann et al., 1974). The relevant components of logging operations are skidding, log landing, and access road construction, where mineral soil may be exposed.

While increases in streamwater nutrients vary by type of cutting and watershed characteristics, the two key aspects of cutting that influence nutrient release are the location and amount of clearing and the response of forest regeneration. Even where openings are revegetated within four years by rapidly growing early successional species, nutrient losses can still occur (Bormann et al., 1974).

Studies have demonstrated the methods that will hold water temperature and turbidity increases within tolerable limits (Swank, 1972). Patric (1978) states that there is overwhelming evidence that neither the productivity of soils nor the quality of water is substantially lessened during or after responsibly managed harvests. Stone et al. (1979) report that if proper precautions are taken, water quality impacts from logging are essentially non-existent. Regarding timber harvesting, Stone (1973) concludes that "adverse impacts can be greatly reduced or entirely avoided by skilled planning and sufficient care."

3.3.3 Uneven-Aged Silviculture

3.3.3.1 Water Yields

While most of the trends summarized in the even-aged management water yields section above also hold true for uneven-aged management, the effects upon water yield vary. For example, uneven-aged management on north-facing slopes, removing conifers and involving significant percentages of basal area, will probably result in higher water yields than less intensive cuts removing hardwoods on south-facing slopes. Either approach to uneven-aged management, however, will likely result in smaller water yields than a comparable even-aged management approach. This is due to less dramatic changes in soil moisture and evapotranspiration caused by the partial cuttings and smaller openings used in uneven-aged management. Adjacent vegetation and advance regeneration more quickly fill these smaller gaps. In addition, adjacent trees utilize part of the additional soil moisture created by cutting. Hunt and Mader (1970) found that when two white pine forest plots at Quabbin Reservoir were thinned by 30% and 80%, soil moisture increased slightly to moderately and growth increased by 70% and 230% respectively. Hornbeck et al. (1993) reported that when 24% of a basin was cut in one clearing it yielded twice the water of a similar basin where 33% of the forest was removed in scattered openings.

Douglass (1983) found that "partial cuttings were not as efficient for augmenting water yield as were complete cuttings." Storey and Reigner (1970) note:

> There are several ways we can manipulate vegetation to effect water savings. The obvious one is by heavy cutting of trees, thereby removing rainfall intercepting surfaces and removing the transpiring agent. According to considerable evidence our people have collected, single tree selection cutting saves little or no water. The cutover area need not be large; cutting in blocks or strips or even group selection of trees to be removed will save water.

While it is clear that silvicultural systems employing partial cuttings yield less water than complete cuttings, partial cutting studies do show increased yields (Mrazik et al., 1980). For example, of the 10 selection cut or thinning watershed experiments in the U.S. listed by More and Soper (1990), 8 resulted in significant yields. The average annual significant yields for each of the first five years after cutting ranged from .4 to 2.3 area-inches. When the ten experiments are averaged, selection/thinning resulted in a yield of 1 area-inch per year for the first five years after cutting. Hibbert (1967) reported results of seven selective cuttings in North Carolina and West Virginia with all watersheds except one having a southerly exposure. The average annual yield for years measured after cutting was 1.13 area-inches. The lightest cuttings necessary to produce significant yields remove approximately 20% of the forest basal area (Kochenderfer and Aubertin, 1975, as cited in More and Soper, 1990; Trimble et al., 1974). Douglass and Swank (1972) assembled a model that predicts a first year water yield increase based on reduction in forest basal area. This model predicts that a 30% reduction in basal area will increase yields approximately 2-3 annual area-inches.

In predicting the significance of water yields to be derived from uneven-aged management, specific site characteristics of watersheds must be examined. For example, cuttings on north facing watersheds with deep soils will result in relatively larger yields. Using regression lines from Hibbert (1967), a one-third reduction in forest cover on a north-facing watershed is estimated to yield three times the streamflow of a similar cut on a south-facing watershed.

Yields from uneven-aged management should also be viewed in comparison to the two above alternatives: even-aged management and natural management. When compared with these two options, uneven-aged management falls between the two. For example, partial clearing of watersheds with even-aged management may yield 5 or more area-inches per year (approximately 25% increase in yield) for the first few years after cutting (estimated from Hibbert, 1967). However, aging forests or naturally-managed watersheds with new forest growth will have reduced water yields over periods without disturbance. For example, Hibbert (1967) reports on three small watersheds (all less than 2,000 acres) in New York where an average of 47% of the watersheds was planted to conifers. After 25 years, the three watersheds averaged 5.3 area-inches less streamflow. Another medium sized watershed (over 300,000 acres) that was passively managed for 38 years and on which average basal area doubled, showed a decrease in yield of 7.7 area-inches - equivalent to a 25% reduction.

Existing data show potential water yield increases of approximately 25% for even-aged management and potential decreases on unmanaged forests of up to 25%. Uneven-aged management falls in between these two approaches, but averages small yield increases (on the order of approximately 5% for the first few years after cutting). The above approximate range would be reduced in actual magnitude depending upon the percentage of the watershed cut and the frequency of the rotation of cuttings. However, the relative comparison of the three alternatives should generally hold true.

3.3.3.2 Water Quality

Many of the principles underlying the potential for water quality impacts as a result of logging operations apply equally to even-aged and uneven-aged management. In order to avoid repetition, only the potential water quality impacts unique to uneven-aged systems will be reviewed in this section. As with even-aged management, the impacts upon water quality vary with several factors: intensity and location of management; intensity, layout, and maintenance of road systems; and planning and supervision of logging and woods roads operations (Lull and Reinhart, 1967; Kochenderfer and Aubertin, 1975; Hornbeck and Federer, 1975).

Uneven-aged systems remove single trees and small groups of trees. In a temperate-region forest study of gap-size impacts on nitrates, Parsons et al. (1994) measured extractable nitrate in soil plots. Within a lodgepole pine forest in Wyoming, gaps were created by removing 1, 5, 15, or 30 trees. The authors found that, compared with adjacent undisturbed forest, gaps created by removing 1 or 5 trees had no increase in nitrate. The 15-tree gaps had higher nitrate levels, and 30-tree gaps had nitrate levels 2-3 times higher than the 15-tree gaps. This same stand was previously thinned with no increase in nitrates, and clear-cut with soil nitrate increases of 10-40 times adjacent undisturbed forest. The authors recommend selective harvesting if nitrogen availability is of concern on a site. Stone (1973) notes:

> Any management practices that reduce vigor of the residual vegetation or delay regrowth and regeneration - such as scarification, excessive herbicide application, or maintenance of excessive deer herds - could increase loss rates [nitrate leaching] above those observed on the harvest clearcuts. On the other hand, greater surface soil shading, as by partial cutting methods, narrow stripcuts, increased cover density on clearcuts, or any means of hastening regrowth, would reduce losses [nitrate leaching] even more.

Trimble et al. (1974), in comparing management systems, state that water quality is ordinarily maximized on forest land by maintaining an unbroken tree and litter cover. The city of Baltimore's forest management utilizes the selection system because "although this [the selection system] is not the most economical system of cutting to use, it leaves sufficient cover to protect the watershed..." (Hartley, 1975).

The literature clearly reports that where stream shading is unaffected, stream temperature will not change (Douglass and Swank, 1975; Hornbeck et al., 1986; Kochenderfer and Aubertin, 1975). With little significant impact upon temperature and nutrient streamwater parameters, the chief potential impact of uneven-aged management systems is turbidity. Increased turbidity appears to be less of a concern with uneven-aged management, however, due to the lighter cutting practices and the amount of forest cover. For example, a comparison study of two watersheds at the Fernow Experimental Forest in West Virginia showed only slight elevations of particulates after three selection cuts during the 1950s and 1960s (cuts included 13%, 8%, and 6% of basal area) as compared to an adjacent undisturbed watershed. In a separate study, Corbett and Spencer (1975) reported no turbidity increases from a thinning operation.

One area of potential concern regarding traditional uneven-aged systems is that cutting cycles are often more frequent, meaning more frequent forest entry and more miles of access roads in use at any given time (Stone et al., 1979). However, the actual impacts will depend upon the uneven-aged method adopted. For example, in uneven-aged forests managed for water supply purposes, trees can be grown on longer rotations and longer cutting cycles. Rhey Solomon, water resource manager for the U.S. Forest Service notes "...the way to keep the water flowing and safeguard the forest is to rotate management throughout the watershed" (American Forest Council, 1986).

3.3.4 Air Pollution Effects on the Forested Watersheds of the Northeastern U.S.

The intent of this section is to look at the impact of air pollution on present and future forests of the southern New England region. For water supply purposes, managers must consider both the forest as an ecosystem and its function as a watershed. The focus must include both the direct impacts of air pollution upon watershed forests and the impacts of resulting ecosystem degradation upon water quality.

While the following discussion outlines specific impacts of air pollution upon forests, it is extremely difficult to isolate these impacts from the many other processes and stresses occurring in forest ecosystems (climatic stresses, insects, diseases, fire, ice, wind, etc.). It is also difficult to isolate the impact of one specific pollutant (e.g., ozone or nitric acid) from the composite of impacts affecting a forest. Klein and Perkins (1988) state:

> It is now recognized that no single causal factor is responsible, but that there are a variety of anthropogenic causal factor complexes interacting with natural events and processes that, together, induce stresses in forests that culminate in declines of individual plants and of ecosystems.

3.3.4.1 Acid Deposition

Carlton (1990) contains an excellent overview of the impact of acid deposition upon watersheds. In Massachusetts, data indicate that the average pH of precipitation is 4.2, which is six times more acidic than uncontaminated precipitation (Godfrey, 1988, as cited in Carlton, 1990). In New England, approximately 60-70% of the acid falls as sulfuric acid and 30-40% as nitric acid (Murdoch and Stoddard, 1992; Rechcigl and Sparks, 1985, as cited in Carlton, 1990). Murdoch and Stoddard (1992) note a study in Maine that showed the sulfuric acid component decreasing in recent years, while the nitric acid component is increasing, leaving the pH of precipitation fairly constant. Stoddard (1991) reported that sulfate deposition had decreased by 1.8% from 1970 to 1984 in the Catskill Mountains of New York. Acidity, however, remained the same due to equal increases in the nitric acid component. In Massachusetts, depositions amount to .3 to .7 pounds of hydrogen ion, 16.2 to 27.5 pounds of sulfate, and 8 to 22 pounds of nitrate per acre per year (Petersen and Smith, 1989).

Sulfuric and nitric acids tend to accelerate replacement of aluminum, calcium, magnesium, and other base cations in the soil with hydrogen ions (Hovland et al., 1980, as cited in Carlton, 1990). In this way, acid deposition will increase soil acidity and directly impact biological activity, soil fertility, and cation-exchange capacity (Carlton, 1990). Acid precipitation can also leach aluminum directly into streams causing potential negative water supply and aquatic and fish impacts (McAvoy, 1989). Key factors in determining the susceptibility of watersheds to acid inputs include: the supply of base cations in soils; the percentage of base-rich groundwater flow versus storm flow; the relative importance of snowmelt events; the average storm rainfall intensity, volume, and duration; and the soil depth, texture, pH, and cation exchange capacity (McAvoy, 1989; Peters and Murdoch, 1985; Veneman, 1984). Records at Hubbard Brook, New Hampshire show that while sulfate inputs have declined, base cation inputs from precipitation have also declined (145 micro eq/liter in 1963 to 104 micro eq/liter 1989) causing sensitivity to acidification to actually increase (Driscoll et al., 1989). The authors attribute the decrease in base cations to a large reduction in suspended particulates since 1970 due to reduction of coal and open burning emissions.

Some researchers have questioned the extent of the impact of acid precipitation. Krug and Frink (1983) feel that most aluminum in streamwater is due to acid soils (caused by natural humic acids) not acid rain. Krug and Frink (1983) and Veneman (1984) note that streamwater can become more acidic as the acid humus layer increases with forest age and because thick humus layers may reduce the amount of water percolating into the subsoil and increase saturated overland flow. Studies in Connecticut and the Berkshires of Massachusetts show that soil acidity increases with forest age (Art and Dethier, 1986; Krug and Frink, 1983). In Connecticut, litter pH changed from 5.5 to 3.9 from 1927 to 1980 and the mineral soil pH from 5.1 to 4.6 during this period. A study in Norway also concluded that changing land use and consequent succession were largely responsible for acidification of soils and water (Krug and Frink, 1983).

Reuss and Johnson (1986) identified the key difference between natural and anthropogenic acid inputs as the ability of the stronger nitric and sulfuric acids to leach through to stream waters, whereas the weaker natural organic acids will leach from upper to lower soil horizons, acidifying soils but not stream waters. Therefore, a key factor in identifying systems acidified by pollution is whether pH is attributed to organic acids or sulfates and nitrates.

Driscoll et al. (1988) noted that the "acid rain" and "acid soil" argument is largely due to the lack of long-term data on basin soil and water quality. To help resolve this controversy, the authors compared two similar basins, one in New Hampshire (NH) where acid deposition is significant (pH 4.1) and one in British Columbia (BC) where acid deposition is insignificant (pH 5.0). The basins have similar bedrock, glacial history, and soils but differed in vegetation type and precipitation amounts. Both headwater streams were acidic. The key difference was that the BC stream was dominated by weak organic acids, had low aluminum concentrations, and low sulfate loading, while the NH stream was dominated by strong acids (nitric and sulfuric), had high aluminum concentrations, and high sulfate loading.

Two streams in the Quabbin watershed, the West Branch of the Swift River and the East Branch of Fever Brook, received similar analysis to those in NH and BC (Rittmaster and Shanley, 1990). The concentrations of sulfate and hydrogen ions in precipitation were significantly higher at Quabbin than at the New Hampshire site. While both Quabbin streams had high aluminum concentrations during high flow periods, Fever Brook aluminum was in an organic form that is not toxic to fish. Fever Brook also had one half the net export of sulfate of the Swift River, a result of sulfate reduction in the extensive beaver flowage at Fever Brook.

Veneman (1984) rated the ability of the soils of Massachusetts to buffer acid inputs using many of the criteria outlined above. Of the 25 soil types that make up almost all of the OWM lands at Quabbin, only four (all wetland soil types) were classified as "acid precipitation will have no negative impact on water quality," whereas sixteen types are listed as "acid precipitation will have a moderate or significant impact on water quality." Baker (1984) re-measured soil parameters at eight sites at Quabbin that had been measured in 1962. He found that soils had increased in acidity and exchangeable aluminum and were now releasing sulfate, whereas they were adsorbing sulfate in 1962. These changes have reduced the neutralization capacity of the soils.

3.3.4.2 Interaction between Air Pollution and Forests

Reuss and Johnson (1986) use the term "canopy leaching" for the process where hydrogen ions replace base cations in the forest canopy. Krug and Frink (1983) report that 90% of the hydrogen ions in acid rain at Hubbard Brook, NH are neutralized in the northern hardwood canopy during the growing season (rain pH of 4.1 changed to 5.0 in throughfall). In studies in the west-central Adirondack Mountain region of New York, Peters and Murdoch (1985) noted that throughfall in deciduous forests was less acid than rain, while thoughfall in coniferous forests was more acid than rain.

As the forest flora exist in several layers above and below the ground surface, the accumulation/neutralization that occurs at these various layers tells a great deal about how the forest processes incoming acid deposition. Yoshida and Ichikuni (1989) studied the chemical changes to precipitation as it passed through the canopies of three different types of Japanese forests. They reported that from 49-74% of the total incoming acid deposition was neutralized by the forest canopies, with deciduous oak forests neutralizing the least and cedar forests neutralizing the most. Virtually all of the cations and anions studied, with the exception of the hydrogen ion, increased as precipitation fell through the canopy (the authors studied Ca2+, Mg2+, K+, Na+, NH4+, H+, Cl-, NO3-, SO42-, and Al). This indicates the process of "canopy leaching" is evident in these forests. The authors note that similar occurrences have been documented in New England by other authors.

Laboratory studies indicate acid precipitation increases leaching of calcium and potassium from vegetative foliage (Smith, 1981). In order for the forest canopy to replace the cations and anions lost, similar amounts of these substances must be taken up from the soil. In some cases, acid conditions cause these nutrients to be leached below the root zone where they become unavailable to plants (Klein and Perkins, 1988). The net effect of the above processes is to acidify the soils and damage forest ecosystems (Yoshida and Ichikuni, 1989).

Increasing acidity of soil water causes leaching of aluminum, an element of increasing concern to water supply managers. Aluminum also damages fine tree roots and inhibits the uptake of calcium, a nutrient vital to plant growth. This situation leads to further imbalance in nutrients and increases susceptibility to drought stress, decline in growth, and increased mortality (Johnson and Siccama, 1983, as cited in Art and Dethier, 1986; Petersen and Smith, 1989; Smith, 1981). For example, soil acidity is a potential contributor to increased nitrate leaching from forests (Vitousek, 1977). Klein and Perkins (1988) report that temperature, moisture, light, nutrients, and soil factors all contribute to susceptibility to disease. This type of pollution may also affect recovery from winter injury.

LOCAL SOURCES OF AIR POLLUTION

According to Klein and Perkins (1988), trees undergoing nutrient stresses may be predisposed to decline when natural and pollution-caused stresses are added. Forests that are damaged by decline go through a process of "reorganization" during which time increased nutrients are leached from the system into tributaries. This increased loss of nutrients may in turn perpetuate the forest decline.

Soil acidity will vary relative to air pollution levels, as well as other factors including soil type and horizon, underlying geology, and successional stage of forest cover (Art and Dethier, 1986). In general, the soils of the New England region have a low acid neutralizing capacity or "ANC" (Godfrey, 1988, as cited in Carlton, 1990). Art and Dethier (1986) studied the relationship of land use and vegetation to the chemistry of soils in the Berkshires. Acidity of the upper-most soil layer was positively correlated to species composition and stand age, with stands less than 140 years averaging pH 4.21 and those over 140 years averaging pH 3.92. Several studies verify an increase in soil acidification with successional sequences following agricultural abandonment (Robertson and Vitousek, 1981, Thorne and Hamburg, 1985, Krug and Frink, 1983, all as cited in Art and Dethier, 1986). Acidity varied with land use history, with previously pastured lands having significantly lower pH in the upper horizons than previously cultivated lands. The conclusion is that past land use has a significant impact on species composition and overall soil acidity (Art and Dethier, 1986). These studies are useful in considering overall differences in chemical processing in various types and ages of forests and in assessing the potential susceptibility of various forests to impacts of acid deposition.

Soil water pH generally decreases deeper into the soil profile. For example, in a study of eight forest soils in central Massachusetts, mean pH in the A and C horizons were 4.39 and 3.58 respectively; an increase in acidity of eight times. Exchangeable aluminum in the A horizons was nearly four times as high as in the C horizons (Baker, 1985, as cited in Carlton, 1990).

High levels of ozone cause injury to leaf surfaces of sensitive tree species such as white pine, black cherry, and white ash, especially during summer months. Ozone also reduces photosynthetic rates and the supply of carbohydrates to the roots (Petersen and Smith, 1989; Reich and Amundson, 1985; Smith, 1981). High levels of ground level ozone occur at Quabbin Reservoir, with readings recorded at Quabbin Hill sometimes exceeding other state recording stations including those in Boston.

The combined effects of acid deposition and ozone pollution may be contributing to a measurable decline in Massachusetts forests. A statewide study of the Massachusetts forests identified 24,000 acres that show signs of decline, including yellowing leaves, dead branches, and standing dead trees. This represents a 10% increase in forest decline over twenty years ago (Parker, 1988). In addition, the growth rate on one third of the red and white pines studied has dropped 20-50% since the 1960s (Freeman, 1987). The overall impact of air pollution predisposes trees to insect and disease outbreaks. Research shows that air pollution predisposes pine trees to bark beetle infestations and makes several tree species more susceptible to root rotting fungus (Smith, 1981).

In Massachusetts, the decline of red spruce and sugar maple has been examined most closely. Studies of red spruce on Mt. Greylock found that this decline involved a combination of factors, including pathogens, insects, and ice, snow, and wind. However, the decline studied was attributable only in small part to these factors. The high acidity of rain and fog, the high soil acidity, and the low soil nutrient content (including low calcium) at these sites point towards air pollution as a chief cause of the decline of red spruce. The study of sugar maple decline also concludes that many trees are in a weakened condition, which magnifies the impact of other detrimental factors (Petersen and Smith, 1989).

In addition to acid deposition and ozone pollution, current air pollution contains metals, polychlorinated biphenyls (PCBs), alkanes, and various polycyclic hydrocarbons and organic acids (Rechcigl and Sparks, 1985, as cited in Carlton, 1990). Soil and vegetation surfaces are the major "sinks" for pollutants in terrestrial ecosystems (Smith, 1981, as cited in Carlton, 1990). For example, the leaves and twigs of an average sugar maple tree 12 inches in diameter will remove the following elements from the air in one growing season: 60 mg of cadmium, 140 mg of chromium, 5800 mg of lead, and 820 mg of nickel (Smith, 1981). Klein and Perkins (1988) report that the accumulation of metals affects nitrogen transformations in hardwood forests.

Forest soils serve as sinks for lead, manganese, zinc, cadmium, nickel, vanadium, copper, and chromium; tree trunks also serve as sinks for large amounts of trace metals including nickel, lead, chromium, cadmium, and manganese (Smith, 1981, Driscoll et al., 1988, as cited in Carlton, 1990). The U.S. Environmental Protection Agency designed a 40-acre "model forest" containing several hardwood species and white pine (Smith, 1981, as cited in Carlton, 1990). The model predicts that, within five years of planting, this hypothetical forest and its soils would annually remove the following pollutants:

96,000.00 tons/year of ozone
748.00 tons/year of sulfur dioxide
2.20 tons/year of carbon monoxide
0.38 tons/year of nitrogen oxides
0.17 tons/year of peroxyacetylnitrate

The net effect of air pollution on a forest ecosystem is a combination of decreased photosynthesis, decreased growth, increased respiration, reduced biomass, and possible reductions in reproduction. These impacts produce a range of symptoms that together are termed "forest decline." The severity of the decline depends on the amount of pollutants, and the species and site conditions involved. An additional impact of air pollution is alteration of forest ecosystem composition and structure, through selectivity of impact. More severe air pollution, and air pollution on naturally stressed sites, serves to simplify the overall make up of the ecosystem and make it less diverse and less stable (Klein and Perkins, 1988; Smith, 1981). Smith (1981) defines three classes of air pollution impacts:

- Class I: low dosage, where ecosystem serves as a sink for pollutants.
- Class II: intermediate dosage causing nutrient stress, reduced photosynthesis and reproductive rate and increased predisposition to insects and diseases.
- Class III: high dosage where mortality is widespread and gross simplification of the ecosystem alters hydrology, nutrient cycling, erosion, microclimates, and overall ecosystem stability.

Klein and Perkins (1988) reviewed more than 400 studies relating to forest decline and concluded:

> There are interactions between primary causal complexes and their direct effects and secondary causes and consequences of forest decline discussed here, so that the web of interactions becomes formidable. Nevertheless, a start must be made on these analyses, not only to understand forest decline holistically, but also because of the pressing need to develop concepts and strategies to ameliorate or reverse the imminent collapse of forested ecosystems. Recognizing that species sensitivities to causal factor complexes vary greatly, inevitable simplification of ecosystems will drastically affect their ultimate stability.

3.3.4.3 **Nitrogen Saturation**

The potential problem of nitrogen saturation, defined as the declining ability of an ecosystem to retain added nitrogen, was only identified in 1981 (Aber, 1992). Researchers are concerned that acid deposition may also be adding significant amounts of nitrogen, originating chiefly from nitrogen oxides in air pollution. The effects of nitrogen saturation include elevated nitrate, aluminum and hydrogen ion concentrations in stream water (Van Miegroet and Johnson, 1993). Monitoring of nitrates is required for drinking water (standard=10 ppm) because of health effects upon infants and potential formation of carcinogenic byproducts (Skeffington and Wilson, 1988). Nitrates can also cause algal blooms in lakes and reservoirs. Excess nitrogen deposition may also effect forest composition and productivity (Aber, 1992).

Bormann and Likens (1979b) report a doubling in nitrate concentration in precipitation since 1955. Schindler (1988) reports that deposition of nitrogen oxides have increased much more rapidly than sulfates in recent decades. Ollinger et al. (1994) report that there is a more than twofold increase of wet nitrate deposition from east to west between eastern Maine and western New York State. The authors mapped broad-scale wet and dry nitrogen deposition across the Northeast, with the Catskill region in the highest category (10.34-12.66 kg N/ha/yr.) and the Quabbin region in the 7.99-9.16 kg N/ha/yr. category.

Processes Involved

The processes related to nitrogen saturation are more complex than those related to precipitation inputs of sulfates, mainly because nitrogen can be both an acid and plant nutrient component and because of the complex interactions between soils and plants and the various compounds of nitrogen. In the ammonium form, nitrogen is a nutrient for the plant/soil biota complex. In the nitrate form, nitrogen can be a nutrient for biota but can also be a very mobile and dominant anion involved in base cation depletion and mobilization of aluminum through the soil and into stream water.

A key reaction in this process is nitrification, the conversion of ammonium to nitrate. Others are denitrification (in which atmospheric nitrogen is released from nitrates) and nitrogen mineralization (the process by which ammonium is formed from organic nitrogen in soils). Mineralization is an important process, as the storehouse of nitrogen in soils far exceeds that in the plant system (75-97.5% of nitrogen is in inorganic form in soils) but the nitrogen can be more mobile in the plant system. As long as the soil system delivers an amount of nitrogen less than or equal to the capacity of the plant system, nitrogen is held within the system. Thus, nitrogen saturation requires both the soil and plant systems to be saturated.

The interaction of these three processes - nitrification, denitrification, and nitrogen mineralization - is dependent upon various bacteria, pH levels, season and climate, as well as variations in plant/soil composition. An added complication is the process of nitrogen fixing, by which plants transform nitrogen gas (the most prevalent component of the atmosphere) to nitrogen in a usable form in the soil/biota system. The relative importance of nitrogen fixation is dependent on the composition of nitrogen-fixing plants in the system. Bormann and Likens (1979b) estimate that 70% of the nitrogen store at Hubbard Brook, NH is derived from fixation and the remainder from deposition. In general, predictions of the timing of the onset of nitrogen saturation are limited by the lack of understanding of soil properties and the complex processes at work there (Schofield et al., 1985; Agren and Bosatta, 1988; Nadelhoffer et al., 1984; Aber, 1992,1993).

Disturbance of the plant/soil system by natural or anthropogenic events tends to increase mineralization of nitrogen and consequent nitrification in the system. Vitousek et al. (1979) analyzed processes that keep nitrate leaching in balance. These include the accumulation of ammonium in soil solution on cation exchange sites in the soil, and lack of soil water for nitrate leaching. A delay in nitrate movement after disturbance is critical as this allows vegetation to develop and take up much of the available nitrate before it can leach into stream waters.

Van Miegroet and Johnson (1993) summarize the complexity of the nitrogen saturation process:

> This soil condition is the integrated result of vegetation type, age and vigor, past N accumulation history, climatic conditions, and current and past N input regime and soil characteristics.

Aber et al. (1989) have developed equations based on field work that can help model the nitrogen cycle using soil litter analysis.

Symptoms and Susceptibility

Aber (1992) describes the characteristics - including annual stream water nitrate trends - of nitrogen-limited, nitrogen-transition, and nitrogen-saturated systems. In general, nitrogen-limited systems have low nitrate loss during snowmelt, high carbon:nitrogen ratios in soil litter, and high soil dissolved organic carbon concentrations. Nitrogen-saturated systems exhibit the reverse conditions for these three criteria. The identification of elevated nitrates in storm events, especially during snowmelt, may be a first indication that system inputs are at least temporarily exceeding capacity. Researchers at the New York City water supply watersheds in the Catskills are concerned about peaks of nitrates in the spring (up to

128 micro eq/l) combined with elevated summer levels (Murdoch and Stoddard, 1992). Rittmaster and Shanley (1990) in studying two tributaries at the Quabbin reported that nitrate concentrations were generally low, but nitrate peaks of 20 and >35 micro eq/l were reported in the two streams during the snowmelt period. The authors attributed these peaks to short soil contact time during storms. There are no other records of nitrate peaks at Quabbin, but limited storm sampling has been done.

Brown et al. (1988) recommend consideration of vegetation type and age, site history, carbon: nitrogen ratios in soil organic matter, external inputs, and nitrogen turnover rates to thoroughly evaluate the condition of a system with regard to nitrogen saturation. The authors note that because natural plant communities change, nitrogen saturation is a "moving target." Van Miegroet and Johnson (1993) reported that forests with small soil nitrogen pools, due to either limited accumulation history or frequent disturbance such as fire, generally have low nitrification potential and insignificant nitrate leaching, irrespective of age or vigor of the forest. Sites that have high soil nitrogen content coupled with a low carbon:nitrogen ratio have a high nitrification potential, and under these conditions the annual leaching of nitrates is strongly dependent on atmospheric inputs, forest age and tree nitrogen uptake rates.

Impacts of Forest Succession and Disturbance

Stand age is an important factor in determining nitrogen uptake and annual nitrogen accumulation rates in tree biomass. A declining trend in nitrogen immobilization as a stand matures may explain why nitrate leaching losses are typically larger in mature versus vigorously growing forests. Long periods without disturbance may allow high nitrogen accumulation and low carbon:nitrogen ratios and increased nitrification potentials (Van Miegroet and Johnson, 1993). Hemond and Eshleman (1984) note that both higher plant uptake and microbial immobilization contribute to limiting nitrate losses from Temperate Zone mid-successional forests. Murdoch and Stoddard (1992) state:

> In watersheds where forests are accumulating biomass, biological demand for nitrogen is often sufficient to retain virtually all atmospherically deposited and mineralized nitrogen during the growing season and reduces net nitrate release to stream water.

In their analysis of elevated summer nitrate levels in Catskill Mountain streams, Murdoch and Stoddard (1992) hypothesize that the older forests in the Catskill Preserve may have a low demand for nitrogen and may therefore be unable to retain all of the atmospheric nitrogen entering the watersheds. The authors are currently engaged in a study of nitrogen cycling in New York City water supply lands in the Catskills, investigating nitrogen input/output in different landscape types and documenting streamwater chemistry changes over short distances (Murdoch, 1993 personal communication).

Aber et al. (1991) note that changes in species compositions may affect the ability of a forest to absorb nitrogen. Pine (due to longer needle retention) takes up less nitrogen than oak or maple. The authors also modeled the timing of nitrogen saturation of a hypothetical forest under different management scenarios. Forest harvesting (removal of nitrogen) slowed the onset of saturation; ozone pollution reduced net primary productivity and moved the onset of saturation up from 300 years in the future (without ozone pollution) to 50 years into the future (with ozone pollution); and alteration of forest species from low nitrogen-demanding to high nitrogen-demanding species delayed the onset of saturation. This modeling exercise did not examine the impact of forest succession.

3.4 Principles of Disturbance Impacts

- Forest overstory blowdown can temporarily increase erosion and nutrient leaching by disturbing soils, increasing decomposition rates, and causing a setback in biomass accumulation rates.
- Severe forest fire can significantly reduce soil infiltration, thereby increasing overland flow of water, sediments, and nutrients.
- A forest that is diverse in age structure and species composition limits the impacts of age- and species-specific disturbances.
- Forests with advance regeneration in the understory will recover more quickly from disturbances to the forest overstory than will forests with poor understory development.
- Younger, shorter trees will sustain less damage from severe wind storms than taller, older trees, due both to their lower tendency to "catch" the wind and stem flexibility.
- While tightly grown, "aerodynamically smooth" stands may deflect wind better than those that are "aerodynamically rough," individual trees that have been grown in more open stands will develop strongly tapered stems that resist wind better than the non-tapered stems of trees grown in tight stands.
- Runoff from infrequent, large storms with associated intense rainfall and flood waters account for much of the annual particulate, sediment, and dissolved nutrient outputs from watersheds.

Air Pollution Impacts

- Forests serve as "sinks" for various environmental pollutants, retaining them and slowing their movement into water supplies. A tall, dense, and layered forest serves this function more effectively than a short, sparse forest.
- Environmental pollution has been linked to general forest decline, which increases the susceptibility of those forests to insects, diseases and other impacts.
- Actively growing forests with a diversity of species and sizes may help buffer the impacts of acid precipitation on water supplies.
- Air pollution contributes to nitrogen saturation of forest ecosystems. Nitrogen saturation can cause elevated nitrate, aluminum, and hydrogen levels in streams and losses of cation bases from soils. These impacts can be compounded by acid precipitation and ozone pollution, and ameliorated by the accumulation of biomass and nutrients in an actively growing forest.

Wildlife Impacts

- Wildlife populations can have significant impacts on both habitat and water quality conditions.
- The composition of wildlife communities is dictated by various factors - chief among them are habitat conditions, landscape characteristics, and mortality factors.
- Land management practices that change habitat conditions will result in changes in the wildlife community.